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NATIONAL MARINE FISHERIES SERVICE

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STATUS OF THE PACIFIC HARBOR SEAL

POPULATION ON THE U.S. WEST COAST
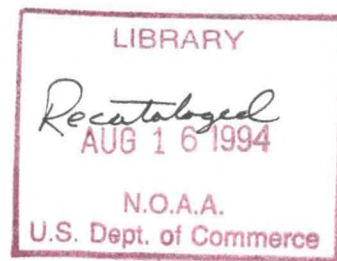
By

Peter Boveng

ADMINISTRATIVE REPORT LJ-88-06

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STATUS OF THE PACIFIC HARBOR SEAL POPULATION
ON THE U. S. WEST COAST

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INTRODUCTION

The harbor seal (<u>Phoca</u> <u>vitulina</u> <u>richardsi</u>) is one of several species of pinnipeds which inhabit the Pacific coast of the United States. Management and conservation of harbor seals are subject to the guidelines and regulations of the Marine Mammal Protection Act of 1972 (MMPA) and subsequent amendments to the Act. Management under the MMPA is premised on determinations about whether the population is in a condition that can be considered optimum (for the animals and the ecosystem) and whether a proposed level of take is detrimental. Consequently, the status of the population must be assessed periodically. This document is a synthesis of the currently available information on the status of harbor seal stocks of the U. S. west coast.

History of Exploitation and Management

Harbor seals have been subject to bounty kills and pelt harvests during this century in various parts of their range. In British Columbia, harbor seals were subject to a bounty kill from 1914 to 1964 and were hunted commercially for pelts after 1962 (Bigg 1969a). The recorded bounties between 1914 and 1964 averaged 2,913 seals annually but Bigg (1969a) suggests that the actual kill was probably about twice that many. He considered 35,000 to be a likely estimate of the British Columbia population size and cited two other estimates of 17,000 and 20,000 seals.

Scheffer and Slipp (1944) estimated that there was a minimum of about 5,000 harbor seals in Washington during the early 1940's. They reported that 3,200 bounty kills were recorded between 1922 and 1926 and suggested that an additional 40 percent of the kills were not recorded. This would imply an average annual kill of about 900 seals during the period. Newby (1973) estimated that 17,133 harbor seals were killed by bounty hunters in Washington between 1943 and 30 June 1960, an average kill of about 1,000 seals anually. Newby (1973) estimated that the Washington population had declined to about 1,700 by 1972.

Pearson and Verts (1970) reviewed the historical records of harbor seal bounties, distribution, and abundance in Oregon. Over 1,000 harbor seal bounties were recorded in 1930. Pearson and Verts (1970) estimated that more than 500 harbor seals were killed annually in the Columbia River between 1938 and 1942. Because of low numbers observed in surveys conducted during 1967 and 1968, and because of dramatic declines in numbers of bounty claims, Pearson and Verts concluded that the number of harbor seals in Oregon had declined substantially since 1930 and that the population was perhaps endangered.

The history of exploitation in California is less extensively documented. Bonnot (1928) observed that during the latter part of

the nineteenth and early twentieth century, harbor seals were generally not abundant enough in California to warrant management legislation or a commercial harvest. He also discussed an influx to California of pinniped bounty hunters from Oregon at about that same time, supposedly a response to declining pinniped populations in Oregon.

Many of the data concerning historical exploitation are anecdotal. However, the statistics cited above from records of paid bounties indicate that harbor seals in British Columbia and the northern states were subject to substantial kills in the early and middle part of this century and that the population declined as a result. Whether numbers in California were reduced by exploitation or limited by natural mechanisms in the early part of this century is not known. One theoretical model (MacCall 1984a) suggests that reduction in numbers near the margins of the range could result from either direct local exploitation or from exploitation near the center of the range. Thus, the recently observed increases in seal counts, to be discussed at length in this paper, may represent recovery from levels reduced by exploitation. However, this explanation is not supported by direct evidence, particularly for California.

Basis For Management (the Marine Mammal Protection Act)

The Marine Mammal Protection Act of 1972 (MMPA) recognizes marine mammals as components of the marine ecosystem. Noting also that some species or stocks of marine mammals have already been endangered by man's activities, the MMPA requires maintenance of stocks above levels at which they might lose their function in the ecosystem. Although ecosystem function forms the basis of the primary objective of the MMPA, marine mammal management in practice is directed toward maintaining optimum sustainable population sizes, a second major objective of the Act.

The working definition of "optimum sustainable population" (OSP) is a range of population size between the environmental equilibrium (carrying capacity, or K) and the level from which maximum productivity would result. The lower limit of the range of OSP is called the maximum net productivity level (MNPL). Implicit in the notion of OSP is the recognition of the instability of population levels below MNPL: that populations reduced to levels below MNPL may decline precipitously, even when subject to apparently moderate perturbations in the environment or the harvest (Clark 1976, Beddington and May 1977). Thus, a population below OSP is designated 'depleted'. On the other hand, populations at levels above MNPL should be more resilient to variable environments and harvests. Population dynamics theory predicts that populations above MNPL will tend to equilibrate with a quota harvest, provided that the harvest is not at a rate greater than the maximum at which the population is capable of growing (Beddington and May 1977).

The net production of the population at the MNPL is the theoretical upper limit to a rate of harvest or incidental take which could be consistent with maintenance of a population within the range of OSP. A level of incidental take, spread proportionally over all age and sex classes, which would cause a population to equilibrate between MNPL and the carrying capacity, is necessarily lower than the rate of maximum net production (MNP).

The present assessment of the harbor seal population uses abundance data, and information about incidental mortality, to estimate the current population size, the current rate of population growth, and to make limited statements about the status of the population relative to OSP.

METHODS OF STOCK ASSESSMENT

OSP Determination

Two basic approaches have been used for determining the status of marine mammal populations relative to the OSP. One approach requires direct estimation of the population sizes which define the range that is considered to be the OSP (i.e., determining the values of MNPL and K). This method has been used for dolphins in the eastern tropical Pacific tuna fishery (Smith 1983) and for harbor porpoise on the U. S. west coast (Barlow 1987). In both of those assessments, the method used to estimate K was a back-projection from current abundance levels to the pre-exploitation level (K), using estimates of the history of the take (Smith and Polachek 1979). Using this method, the OSP determination is relative to the maximum population level that the environment could support at the beginning of the exploitation period (K is an estimate of "historic K"). This method is not applicable to harbor seals because the population may be recovering from a previously exploited state. Thus, a back-projection would require knowledge about the level to which the population was reduced, as well as quantitative estimates of the take which caused the reduction (neither of which is available).

The other approach to stock assessment depends on inference from some index which varies in a predictable fashion with population status (e.g., growth rates or other vital rates, physical growth rate or condition, parameters such as age at sexual maturity or age at first reproduction). Then the value of the index is used to determine qualitatively whether the population is depleted or in the range of OSP, without actually determining the population sizes which define the range. This approach makes an OSP determination that is relative to the current condition of the environment. Dynamic response analysis (Goodman, in press), which was previously applied to California

sea lions (DeMaster et al. 1982) and to elephant seals (Boveng et al. in press), is a form of this method which uses density dependent changes in the population growth rate (indexed by changes in pup counts) to infer the qualitative status relative to MNPL. The form of the dynamic response analyses conducted on elephant seal and California sea lion populations depends on the absence of a significant harvest or incidental take. Because the estimated recent levels of harbor seal take are substantial, application of the dynamic response method would require use of the estimated levels of incidental mortality. This introduces an additional complication because correction of the per-capita production for the harvest requires that the abundance estimates be in terms of total numbers, not simply an index such as peak counts (Goodman, in press).

Estimation of Rates of Increase and Replacement Yields

Rate of change in population size can be estimated directly from trajectories of a population index, or census, through time; or it can be estimated indirectly from estimates of survival and reproduction rates. The former method is an estimate of the actual population growth rate which occurred on the chosen time interval. In the absence of information about trends in population size, the latter method can be useful for determining the growth rates which would be expected on the basis of observed or assumed life history parameters of the species. These values may then be compared to estimated or proposed rates of take to predict the degree of impact on the stock. In the present case, trends in counts are assumed to be at least as reliable as harbor seal life history data (e.g., Bishop 1967, Bigg 1969a, Pitcher 1977, Boulva and McLaren 1979). Therefore, the sections entitled POPULATION GROWTH RATES and REPLACEMENT YIELD are based on analysis of trends in counts.

STOCK IDENTITY

Although the taxonomy of the genus Phoca in the North Pacific has at times been confused, recent clarification by Shaughnessy and Fay (1977) established that the harbor seals found on the coasts of Mexico, the continental U. S., and British Columbia are all properly called Phoca vitulina richardsi. However, there is still some uncertainty about the distinction between P. v. richardsi and P. v. stejnegeri which may overlap and interbreed in the Aleutian Islands (Shaughnessy and Fay 1977). This assessment will be primarily concerned with Phoca vitulina richardsi on the U. S. coast between Canada and Mexico.

In an assessment of harbor porpoise stock status in central California, Barlow (1987) followed the definition of stock used by Larkin (1972) and MacCall (1984b): A stock is a collection of

animals that can be sensibly managed as a single unit. The aspects of harbor seal biology and management which are relevant to applying this definition of stock identity are described below.

Until recently, harbor seals were generally considered to be non-migratory, with persistent major haul-outs and breeding grounds (Scheffer and Slipp 1944, Stoel 1981, Bigg 1981). Several studies, discussed below, observed geographic clines, or differentiation, in pelage color, pupping date, and pesticide residues.

Shaughnessy and Fay (1977) compiled data from the literature on frequency of light and dark phases in adult pelage, revealing a cline along which pelage is generally darker to the south. Bigg (1969b) suggested that mean pupping dates exhibited clines toward later dates as latitude of the pupping site approached that of northern Washington from either direction. However, Temte (1986) offered a simplified version in which the mean pupping date on the outer coast is progressively earlier to the south, while pupping in the Puget Sound area occurs separately, about three months after coastal pupping at similar latitude. Bigg (1973) found variation in onset of estrus in females from two locations in British Columbia and one in California and presented evidence that the variation was under genetic control. Calambokidis et al. (1985) found that concentrations of PCBs, DDE, and their ratios were different in animals collected from Puget Sound and from the outer Washington coast. Taken together, these studies suggest that in some areas, mixing is sufficiently limited to maintain clines and possibly genetically isolated demes.

On the basis of the work by Temte (1986) and Calambokidis et al. (1985), I have assumed for this assessment that the harbor seals in the inland waters of Washington (northern Puget Sound, Strait of Juan de Fuca, southern Puget Sound, and Hood Canal areas) compose a single stock, separate from the outer coast. (Note that Jeffries[1] has observed differences between dates of peak pupping in the areas listed above and recommends that they be considered separate stocks and that any quotas should be established separately for each area. I have not divided the stock that finely, in part because the data are already too sparse.) Information on movements of tagged seals, location of major haulouts and rookeries, and locations of fisheries with most potential for interactions are considered below in dividing the coastal population into discrete stocks.

Despite the evidence cited above in support of limited exchange between some local populations of harbor seals, there is a steadily increasing collection of observations of movements from tagged seals. Recent studies have documented use of several

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[1]Personal communication. S. Jeffries, Washington Department of Wildlife.

different haul-outs, sometimes separated by more than 100 km, by individual seals in a single season (Pitcher and McAllister 1981, Stewart and Yochem 1983, Jeffries 1985, Brown and Mate 1983, Allen et al. 1987, Herder 1986).

In Washington and Oregon, harbor seals use some estuaries and bays for breeding and others primarily for feeding, moving between prefered sites as food availability or reproductive seasonality necessitates (e.g., Brown and Mate 1983, Jeffries 1985). In particular, Jeffries (1985) and Brown (1986) found that the lower Columbia River was used extensively for winter feeding by harbor seals which breed, give birth, and molt in adjacent Washington and Oregon bays and estuaries. This overlap in the lower Columbia mirrors the use of the area by Washington and Oregon salmonid and sturgeon gill net fisheries which have the greatest potential for interactions with harbor seals. Census coverage by Washington Department of Wildlife (Jeffries 1986) and by Oregon Department of Fish and Wildlife (Brown 1986) has also overlapped in the Columbia River area. On the basis of these studies, I have assumed that the harbor seals of the outer coasts of Washington and Oregon compose a single stock.

The California fisheries most subject to interactions with harbor seals are the set gill net and drift gill net fisheries which are limited to waters south of the Russian River (except a small gill net fishery operated by Indians at the Klamath River). Thus, the concentrations of gill net activity with most potential for interactions with harbor seals, the Columbia River and California south of the Russian River, are separated by about 900 kilometers. There are several known movements of seals between Oregon and California. One subadult female moved over 300 km from the Klamath River, California to Alsea Bay, Oregon and in the same study a subadult male moved over 150 km from Klamath River to Rogue Reef, Oregon (Herder 1985). A flipper tag from a seal tagged at Netarts Bay, Oregon was recovered in a herring net in Humbolt Bay, California, 550 km to the south (Brown and Mate 1983). However, these interstate movements appear to be much less frequent than shorter range movements within state boundaries observed in the same studies (perhaps a result of more resight effort near the study site). Recent harbor seal censuses conducted by Hanan et al. (1987) of the California Department of Fish and Game (CDFG) and by the Oregon Department of Fish and Wildlife (Brown 1986) have been limited to their respective state boundaries.

Because of the discreteness of census effort, and the geographically distinct areas of concentrated incidental mortality, I have assumed that the harbor seal population on the coast of California is a separate stock from that of Oregon and Washington. The extent to which movements occur between those states has not been sufficiently studied to use as a basis for stock identity, but the limited evidence available does not indicate that those movements are common.

The majority of seals counted in the Southern California Bight (SCB) are at the Channel Islands and therefore may experience somewhat different environmental conditions and levels of disturbance than seals in central and northern California. The degree of isolation from the mainland of the harbor seals which breed on the islands of the SCB is uncertain. Yochem and Stewart (1987) reported resights of several animals on the mainland after tagging them on San Nicolas and San Miguel Islands. However, sample sizes are as yet insufficient to estimate rates of exchange between the mainland and islands. There are no known records of individual harbor seals born on the mainland but breeding on the islands or vice-versa; nor is there any record of an animal changing its breeding locale from island to mainland or the reverse (but there has been no thorough, long-term study of the issue). I have not assumed a separate stock in the SCB, but in the sections on incidental mortality, and on stock status, the implications of considering the harbor seals found south of Point Conception as a separate stock are discussed.

ESTIMATION OF CURRENT POPULATION SIZE

Regional Censuses

Harbor seal counts have been made using a variety of ground and aerial survey techniques. Early surveys (e.g., Bonnot 1928, Carlisle and Aplin 1966, Scheffer and Slipp 1944) tallied harbor seal numbers incidentally to censuses directed more toward other species and/or did not take into account the variation in hauling patterns typical of harbor seals. More recent surveys have been conducted with greater attention to factors such as season, tide height, time of day and disturbance, which affect the numbers of seals hauled out at the time of census (e.g., Stewart 1982, Miller et al. 1983, Allen et al. 1984, Hanan et al. 1985, Brown 1986, Stewart et al. in press).

Counts obtained in surveys designed to minimize or eliminate the possibility of double counting or undercounting are used here as minimal estimates of population size and as indices of trends in abundance. California and Oregon have been surveyed on a statewide basis; surveys in Washington have usually been confined to specific geographic regions based on timing of pupping and scientific interest, sometimes overlapping areas in Oregon. Current abundance is considered separately for each stock prior to estimating total U. S. west coast abundance. A detailed discussion of trends is deferred until the section entitled POPULATION GROWTH RATES.

1) Washington (inland waters)

The Washington Department of Wildlife (WDW) has annually surveyed the Strait of Juan de Fuca, San Juan Islands, and Puget

Sound (including Skagit, Padilla, Samish, Bellingham and Boundary Bays) since 1984[2]. On 23 and 24 August 1984, an estimated 4,522 harbor seals, including 481 pups were counted in those areas[2]. Calambokidis et al. (1985) counted 719 seals in southern Puget Sound on 9 September 1984 and 821 seals in Hood Canal on 13 September 1984. These counts are combined with the WDW counts and used here as the minimum population estimate of 6062 harbor seals for Washington's inland waters.

2) Oregon and Washington (outer coast)

Brown (1986) reported aerial photo censuses from the Oregon coast from summer months for the years 1977-1984 (Fig. 1). The highest count, 3,825 harbor seals, was obtained in June, 1984.

Harbor seal censuses in Washington have been designed to survey areas of particular scientific or management interest rather than being delimited by state boundaries. The most extensively surveyed areas are the Columbia River with several bays and estuaries to the north (and to the south in Oregon) (Jeffries 1986, Johnson and Jeffries 1983), and the inland waters of Washington including Puget Sound and Hood Canal (Calambokidis et al. 1985). Because Brown's (1986) censuses of the Oregon coast include the Columbia River, only censuses of Willapa Bay and Gray's Harbor (Fig. 1) are used here from Jeffries (1985) and Johnson and Jeffries (1983).

The high count in Willapa Bay and Gray's Harbor was in 1982 when a total of 5,869 harbor seals were counted (Jeffries 1985), but counts more recent than 1982 were not available. In addition to the census areas represented in Figure 1, Johnson and Jeffries (1983) counted 1,456 seals on the northern outer coast of Washington in June of 1977. Jeffries[2] counted 2,103 seals inhabiting the northern outer coast (Olympic Peninsula) of Washington during an aerial survey on 8 September 1978, and this count is used for analysis.

3) California

Surveys for harbor seals on the mainland coast of California have been conducted by Carlisle and Aplin (1966), Frey and Aplin (1970), Carlisle and Aplin (1971), Mate (1977), Bonnell et al. (1983), Miller et al. (1983), and Hanan et al. (1985, 1986a, 1986b, 1987). Those counts are represented in Figure 2. Surveys at the Channel Islands were conducted by Odell (1971), Bonnell et al. (1980), Stewart et al. (in press), and by Hanan et al. (1986a, 1986b, 1987). Counts from surveys of San Clemente and Santa

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[2]Personal communication. S. Jeffries. Washington Department of Wildlife.

Catalina Islands were provided by Oliver[3]. The Channel Islands counts for 1981-1986, shown in Figure 2, were obtained as follows: The 1981 and 1982 counts are from Stewart et al. (in press). The 1983 count was estimated by adding Stewart et al.'s (in press) counts for all islands except San Clemente and Santa Catalina to counts from the latter two islands (Oliver[3]). The 1984 count is from Hanan et al. (1986b). The 1985 and 1986 counts were obtained by adding San Clemente and Santa Catalina counts from Hanan et al. (1986b and 1987) to counts from the remaining islands by Stewart et al. (in press).

The broken lines in Figure 2 signify that the techniques used to obtain the connected counts may not be directly comparable. The CDFG counts for the mainland, and the recent counts for the Channel Islands, are connected by solid lines because survey techniques were consistent from year to year. The 1986 California mainland count was 13,913 in 1986 (Hanan et al. 1987). The 1986 Channel Islands count was 3,887 harbor seals (Stewart et al. in press, Hanan et al. 1987).

Minimum Population Estimate From Counts

A minimum estimate of the number of harbor seals on the U.S. west coast was constructed by combining counts (Table 1) from coastal California, the Channel Islands, Oregon and Washington (including Puget Sound). The combined minimum population estimate for all stocks is 35,659 harbor seals.

Correcting for Seals Not Hauled During Census

Several studies of harbor seal hauling behavior have been conducted for the purpose of estimating what proportion of the population is represented on shore (Pitcher and McAllister 1981, Stewart and Yochem 1983, Herder 1985, Yochem and Stewart 1985, Allen et al. 1987, Harvey 1987, Yochem 1987, Yochem et al. 1987). These studies used radio-tagged seals to detect the proportions of the tagged sample which were on shore at various times or to determine the proportion of days on which individual tagged seals hauled out. Although the method of using radio transmitter tags was common to all studies, the methods of detecting hauled seals and estimating hauling proportion varied widely as did several factors such as geographic location and season. Due to various constraints, discussed below, none of the studies was able to directly estimate the proportion of the population which would be hauled during a regional census at peak molt.

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[3]Personal communication. C. Oliver, National Marine Fisheries Service, Southwest Fisheries Center, La Jolla, CA.

Here I provide a review of the tagging studies, with particular reference to aspects of use in correcting counts. First, I briefly summarize the reported results from the existing studies. Then I describe some of the problems encountered in correcting a regional census from the radio-tag studies. Finally, I present approximate correction factors for use in converting the existing counts to estimates of total numbers.

1) Summary of radio-tag studies

Pitcher and McAllister (1981) radio-tagged 35 harbor seals (24 mature females, 5 immature females, 5 mature males, 1 immature male) at a major haulout on Tugidak Island, Alaska, between 8 May and 9 July 1978. They monitored 7 seals (age and sex composition not given) which they considered to be residents of the southwestern hauling area of Tugidak Island, Alaska for 25 days in June, 1978. Based on the number of days during which each individual seal was observed hauled out at the daily peak, an estimate of 0.50 was computed as the proportion of the population which the daily peak count represents. Pitcher and McAllister (1981) also searched by aircraft for tagged seals which were using other haulouts in the region. They monitored 12 seals (age and sex not given) which they considered residents (not found using other haulouts) during the period 1 August - 5 September. These 12 seals hauled out on an average of .41 of the 31 days observed.

Stewart and Yochem (1983) radio-tagged seals at San Nicolas Island and monitored hauling behavior during May, June and July of 1982. Monitoring was done manually during afternoon peak haulouts. Radio-tags were placed on 4 adult females, 4 adult males, and 2 subadult males. They reported that the tagged seals hauled out on about 65% of the days in May, 58% of the days in June, and 41% of the days in July. They also presented the proportion of tagged seals observed hauled each day, averaging those values within months (these two methods of computation yield the same result when a fixed group of tagged seals are all monitored on the same days).

Yochem et al. (1987) radio-tagged 18 seals (1 adult male, 9 juvenile males, 4 male pups, 1 adult female, and 3 juvenile females) at San Miguel Island and monitored in October, November and December of 1982. Automated monitoring was conducted 24 hours per day. One seal was never resighted. In that study, tagged seals hauled out on 37% of the days between 24 October and 3 December. Expressed in terms of the average daily proportion of the tagged sample which was observed to haul out during that period, the value is 41%. In addition to proportion of days hauled by individual seals, and proportion of tagged seals hauled on individual days, they were able to compute the proportion of tagged seals which were hauled during 1 hour intervals, around the clock (Yochem et al. 1987). The intervals with the highest average proportions hauled out (.19) were 1300-1400 and 1400-1500.

The intervals with the lowest proportion hauled (.11) were 1900-2000, 2100-2200, 2200-2300 and 2300-2400.

Yochem and Stewart (1985) presented results from a radio-tagging study at San Nicolas Island in 1983. Their methods were similar to those in Yochem et al. (1987). They found that contrary to their expectations, hauling proportions were lower in June (19% hauled on an average day), than in March (66%) and April (66%). Percentages in May, July, August, and September were intermediate. They attributed the unexpected results to small sample sizes, erratic transmitter function near the end of the study, and possibly to effects of El Niño on harbor seal feeding and hauling patterns. Therefore, I did not include results from Yochem and Stewart (1985) for estimating population sizes.

Herder (1985) attached radio-tags to 12 harbor seals (4 adult males, 3 subadult males, 1 male pup, 1 adult female, 3 subadult females) in the Klamath River estuary between 8 June and 14 October 1982. Based on observations of 5 seals (age and sex not given) which were considered to be resident in April and May, Herder estimated that seals in the study area hauled out on 56% of the days in April and 62% of the days in May (because of the small sample, standard errors were about one-half the magnitude of the means). The average proportions of the sample which hauled out were 56% and 65% in April and May, respectively.

Allen et al. (1987) radio-tagged 17 (9 male, 8 female) harbor seals in late July (after peak molt), 1985 at Drakes Estero, California. That study classified 8 seals (5 males, 3 females) as resident to the Drake's Estero hauling area. The average proportion of resident seals hauled at the daily peak (0800-1000 and 1000-1200) in August was 0.7.

Harvey (1987) radio-tagged 26 (4 male, 22 female) seals at Alsea and Yaquina Bays, Oregon, in 1983 and 1984. He monitored presence of the tagged animals during 97 (approximately weekly) ground count censuses of 10 haulout sites between Siletz Bay and Strawberry Hill in 1983-1985. The censuses at each site were of short duration (5 to 10 minutes). For a particular census, he calculated the proportion of tagged seals hauled (PTSH) as the number of tagged seals that were ashore, divided by the total of tagged seals located during that census (tagged animals in the water could be identified by the continuity of the radio signal). He computed the mean PTSH for each calendar month and used the inverse of the monthly mean to correct all the censuses within that month. Monthly mean PTSH ranged from 2.9% in November to 82.5% in June (100% in July, but only 1 count was made on which 1 seal was located).

2) Obstacles to correcting counts

Estimation of hauling proportion is a variation of mark-recapture techniques for closed populations. There are several

obstacles to applying the results of these studies to a direct correction of a regional census, nearly all of which are related to a failure to satisfy the standard assumptions of closed population mark-recapture methods (Seber 1982).

First, because California censuses are conducted at the time of peak molt, when tags glued to the pelage do not adhere reliably, some of the radio tag results are from months other than the census month (censuses usually in May or June). Correcting an independent census made in another season presents the possibility of violating the closure assumption. Because of the strongly seasonal pattern of on-shore abundance of harbor seals, estimates of hauling proportion not made at the same time as the census can not be used directly to correct the counts to absolute numbers.

On a finer time scale, problems arise from the nature of the proportion-hauled estimate which is usually an integral over time, complicating comparisons with an instantaneous count from an aerial photo. As and Yochem and Stewart (1985) and Yochem et al. (1987) showed, the proportion of tagged animals which haul in a particular 1 or 2 hour window may be much smaller than when the window is an entire day.

Because of limited resources, the radio tag studies have usually been limited to intensive resight effort at one or a few sites, with occasional surveys of more distant sites for seals which may have relocated. Seals which move from the primary site complicate the choice of a denominator for the hauling proportion estimate. The appropriate choice depends on the relative rates of immigration, emigration and mortality. Unfortunately, those rates are generally unknown for the harbor seal populations which have been radio-tagged.

Perhaps the most serious problem for estimating hauling proportion or total abundance is the existence of age- and sex-specific hauling patterns. If the animals ashore are not representative of the age and sex structure of the total population, most capture techniques will result in a biased sample. Furthermore, if the age or sex structure changes between the release of tagged animals and the recapture, the equal resight probability assumption is violated. This implies a potential for very large errors. In fact, age- and sex-specific hauling patterns have been observed (Stewart and Yochem 1983, Allen 1985, Yochem and Stewart 1985, Allen et al. 1987, Yochem 1987, Yochem et al. 1987). Yochem (1987) reported that the period of peak on-shore abundance (late May - early June) at the Channel Islands occurs when females are nearing the end of the molt and males are just beginning to molt. Thus, a mark-resight effort just before the peak might capture and mark mostly females. If the resighting (radio telemetry) occurred after the peak, small fractions of tagged females hauled concurrently with large numbers of untagged males might underestimate the proportion hauled (overestimate of total abundance). Other scenarios are possible. Thus, the

magnitude, and even the direction of error could be very sensitive to the timing of census events within the rather narrow time interval (about 2 weeks).

Another type of heterogeneity in hauling patterns is the tendency for some harbor seals to haul out at night (Miller 1983, Allen et al. 1987, Yochem 1987, Yochem et al. 1987, Yochem and Stewart, in press). Yochem (1987) suggested that diel patterns in abundance may be a result of overlap between hauling bouts of seals which haul mostly at night and seals which haul mostly during the day. If a study population of seals is segregated into day-hauling and night-hauling groups, relative timing of capture and resight effort could potentially affect the estimate of hauling proportion.

3) Approximate correction factors for this study

The difficulties described above suggest that predicting the direction (let alone the magnitude) of probable errors in the correction factors estimated in individual radio-tag studies is not possible at this time. However, for the purpose of providing a range of population estimates bounded below by the count and above by some corrected value, I have assumed that the estimates of hauling proportions from radio-tag studies considered as a group indicate the approximate magnitude of the correction factor.

The observed sample hauling proportions from the studies cited above, have typically fallen in the range of 50 to 70 percent (Pitcher and McAllister 1981, Stewart and Yochem 1983 - San Nicolas Island, Herder 1985, Allen et al. 1987). At San Miguel Island, Stewart and Yochem (1983) observed mean diurnal peak hauling proportions of 19 percent. The latter estimate was from late fall and early winter, when the abundance ashore at the Channel Islands might be only about 20 to 30 percent of the early summer peak (Stewart and Yochem 1984a, 1984b). Adjusting their estimate of 19 percent by say, a factor of 4, would correspond to a 76 percent estimate at peak seasonal on-shore abundance.

There is little evidence to suggest that censuses obtained during the peak molt by aerial survey techniques detect less than 50 percent of the total population. Therefore, I have assumed that the population is not likely to be more than a factor of 2 larger than the count. The evidence cited above from radio-tag studies, to the extent that the assumptions hold, suggests that counts over a broad geographic range might reasonably be expected to represent about 70 percent of the total population. The correction factor would then be about 1.4. At the other extreme, the counts themselves (factor of 1) are a minimum estimate of population size. These three methods of estimating the population size will be included in the status assessment, with the intermediate estimate (factor of 1.4) considered most likely. Table 2 shows the counts and total population estimates for each stock using corrections factors of 1.4 and 2.0.

POPULATION GROWTH RATES

Inland Washington waters

Calambokidis et al. (1985) provided several estimates of rates of increase at individual study sights in Southern Puget Sound, Hood Canal, and north of Puget Sound. They computed a "growth index" based on three indices measured in 1977-1979, and again in 1984. The three indices were annual high count, mean of daily peak counts, and counts of pups born. The average of their growth index values, weighted by their 1984 mean of peak counts, represents a 13.9 percent annual increase.

Oregon and Washington

Harvey (1987) estimated that the Oregon population increased at 6 percent annually during the years 1975-1984. Brown (1986) considered several related indices (total count, non-pup count, and June non-pup count) from statewide Oregon counts made during summer months in 1977-1984 and estimated population growth rates of 6 to 8 percent annually. Jeffries (1986) estimated the rate of increase in pup counts from the Columbia River, Willapa Bay and Grays Harbor to be 19.1 percent annually and the rate of increase in the non-pup counts from the same region to be 10.7 percent annually, between 1976 and 1982.

California

Population growth rates were estimated by linear regression of the natural logarithm of counts versus year. For mainland California, the curve in Figure 2 represents an average annual rate of increase of 14.7 percent from 1965 to 1986(slope=.137, P<.001). Prior to 1983, the counts increased by 15.7 percent annually (slope=.146, P<.001). There is no significant trend (slope= .028, P>.5) in the logarithm of counts from the years 1982-1986, which were collected by consistent methods by California Department of Fish and Game. However, the count obtained for 1982 (Miller et al. 1983) deviates substantially from the otherwise smoothly increasing trajectory. The Niño event which occurred in 1983 (Barber and Chavez 1983) complicates the interpretation of the high 1982 count. If the 1982 count was unusually high because of some error in the survey or because of some unknown factor which caused an unusually high proportion of the seals to be hauled out, the 1983-1986 counts could easily be interpreted as consistent with the trend prior to 1982. On the other hand, if the 1982 count reflects a genuine increase in the population, the decrease exhibited by the 1983 count and subsequent resumption of growth would suggest that the population actually declined during El Niño or that El Niño had a persistent effect on the proportion of the population which hauls out during the census.

Examination of the Channel Islands counts for 1981-1983 (Fig. 2) suggests that the high 1982 mainland count is genuine. Those Channel Islands counts (Stewart 1982, Stewart and Yochem 1984a, 1984b) were collected independently of the mainland counts but both series show the same pattern. Also, the number of occupied hauling sites and the mean number of seals per site (Fig. 3) for the 1982 survey were not very different from the same statistics for more recent surveys (Hanan et al. 1987).

On the Channel Islands, the apparent trend in counts prior to 1982 was consistent with the observed increase on the mainland (Fig. 2). The population growth rate estimated by regression for the years 1975-1986 at the Channel Islands was 9.1 percent (slope = .087, P<.001). There is no significant trend in the counts from 1981-1986 (slope = .018, P>.5).

The apparent increasing trend in the counts for both the mainland and the Channel Islands prior to 1982 may be due in part to increasing effectiveness of the survey techniques used. Early censuses (Carlisle and Aplin 1966, Frey and Aplin 1970, Carlisle and Aplin 1971) were opportunistic counts of harbor seals made during surveys designed to count California sea lions. Later censuses (e.g., Stewart 1982, Miller et al. 1983, Hanan et al. 1985, Hanan et al. 1986a, 1986b, Hanan et al. 1987) made substantial efforts to survey during peak periods of on-shore abundance. However, it is unlikely that the entirety of the apparent increase in counts is due to increasing survey efficiency, so the series of counts prior to 1982 indicates a recovering population.

INCIDENTAL TAKE

Harbor seals are subject to several types of non-natural mortality. Unknown numbers are shot illegally to protect fishing gear or catch and to reduce numbers. Also unknown is the number entangled in marine debris (Stewart and Yochem 1985), which includes discarded fishing gear. The remainder of this section focuses on quantitative studies of the numbers killed incidentally in fishing operations.

Washington (inland waters)

Jeffries[4] estimates that 50 to 100 harbor seals are taken ncidentally in Puget Sound gill net fisheries each year. Those estimates are equivalent to .82 to 1.6 percent of the minimum stock size estimate, .59 to 1.2 percent of the intermediate stock

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[4]Personal communication. S. Jeffries, Washington Department of Wildlife.

size estimate, and .41 to .82 percent of the maximum stock size estimate.

Oregon and Washington (outer coast)

Geiger (1985) reported on an observer program and estimated harbor seal mortality in the Columbia River for the years 1980-1982. The estimated numbers of harbor seals killed in those years were 193, 334, and 210, respectively (the 1982 estimate was based only on the Columbia River winter gill net season). I have assumed that the mean of those estimates, 246, represents the annual harbor seal mortality incidental to Columbia River gill net fisheries in the early 1980's. Brown[5] and Jeffries[6] suggest that the increased fishing effort (gill net-days), allowed in recent years due to rebuilding Columbia River fisheries, may result in additional kills of 120 to 150 seals annually. Thus, I have assumed that the total annual mortality in the Columbia River gill net fisheries is about 350 to 400 seals. Willapa Bay and Grays Harbor had estimated takes totaling 142 seals in 1980 (Geiger 1985). If mortality in those areas has remained constant, a total estimate for the Washington-Oregon outer coast stock would be about 500 harbor seals killed annually. That value represents about 4.2 percent of the minimum, 3.0 percent of the intermediate, and 2.1 percent of the maximum stock size estimates, respectively.

California

In California, the CDFG has placed observers aboard commercial fishing vessels in the southern California drift gill net fishery (e.g., Diamond et al. 1986a, 1986b), the southern California nearshore gill and trammel net fisheries (Collins et al. 1984, 1985, 1986), the north-central California gill and trammel net fisheries (Wild 1985, 1986), and the south-central California gill and trammel net fisheries (Haugen 1986). From these observations, estimates of harbor seal mortality for California have been produced (Hanan et al., in prep.).

The 1985 estimate for the entire state of California, based on a bootstrap resampling of the CDFG observed fishing effort and on estimates of total effort from landing receipts and skippers' logs, is 1,849 seals killed (Hanan et al., in prep.). In Table 3, that kill estimate is broken down by strata used for the bootstrap. The distribution of the bootstrapped 'replicates' is shown in Figure 4. The central 90% of the distribution lies

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[5]Personal communication. R. Brown, Oregon Department of Fish and Wildlife.

[6]Personal communication. S. Jeffries, Washington Department of Wildlife.

between 1,410 and 2,250. The central estimate and the confidence limits are shown as percents of the estimated 1985 California stock sizes in Table 4, using the range of abundance correction factors presented earlier.

Expressing the estimated incidental mortality as a percentage of the estimated stock size ignores information about the age structure of the kill. Wild (1985, 1986) reported that most of the harbor seals which were observed caught in gill nets in central California were small, probably young-of-the-year. This raises the possibility that if large kills of young seals have occurred only recently, the effect on the population may not be revealed in the counts until the affected cohorts have recruited to the mature fraction. The timing of the census (May or June) relative to the majority of the central California mortality (July and August) suggests that the loss of a major portion of the pup cohort would not be detected as a lack of pups in the census photos.

Pup production is difficult to measure in this species, but several studies have reported values of between 20% and 25% of the non-pup population size or population size just prior to pupping (Bigg 1969a, Brown and Mate 1983, Calambokidis et al. 1985). If those values apply to California, pup production estimates would lie between 2,967 and 7,120 pups annually, depending on which population correction factor and pup production value was used. The central value, 4,577 pups corresponding to pup production of 22.5% and a population correction factor of 1.4, is about 2.5 times as large as the total kill estimate. Thus, if the kill were entirely composed of pups, 40% of the cohort would be taken. Predicting the impact of such a take would require knowledge about the timing of the incidental take relative to periods of high natural pup mortality.

The variation in the bootstrap 'replicates' is strictly a function of the distribution of numbers of seals killed in each observed net pull. The effort estimates made by CDFG also have variance estimates associated with them, but that variation was not included in the bootstrap technique. Because the effort estimates are multiplied by the mean kill rate in each bootstrap replicate, and because the kill and effort data are divided into 13 strata, it is unclear how much the actual variance of the estimated total kill would be inflated by the variance in the effort estimate.

Hanan[7] has computed preliminary estimates for harbor seal mortality in central California gill net fisheries for the years 1983-1984. (These estimates are simple ratios based on observed effort and observed rate of kill. They are subject to change when

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[7]Personal communication. D. Hanan, California Department of Fish and Game.

more detailed stratification methods are used.) The estimate for central California in 1983 was 567 seals killed (s.e.=202). For 1984 the estimate was 857 (s.e.=127). The central California portion of the bootstrapped estimate for 1985, was 1,100 seals killed. Thus, the estimated central California gill net mortality may have nearly doubled between 1983 and 1985.

To estimate incidental fishing mortality in central California for years from which no observation-based estimate was available, I parameterized the relationship between halibut landings (estimated by number of landing receipts) and the estimated kills in 1983, 1984 and 1985, in the same manner as Barlow (1987). Numbers of landing receipts attributable to set-nets were estimated as the number reported caught with entangling gear plus a fraction of the number of receipts reporting unidentified gear. The fraction used was computed as the ratio of receipts associated with entangling gear to receipts associated with all types of identified gear in the same year. However, Barlow (1987) suggested that gear reporting was biased in 1983 and 1984 because the political climate was unfavorable for gill netting. Therefore, the fractions used to prorate unidentified gear in those years were taken from the adjacent years (i.e., 1982 fraction used to prorate 1983 unidentified gear and 1985 fraction used for 1984 gear).

The parameterization consisted of a regression of the kill estimates (dependent variable) on the numbers of landing receipts (Fig. 5). The regression was forced through the origin, on the assumption that zero fishing effort would result in zero incidental kills. The regression was repeated in a bootstrap fashion, sampling many times from the estimated distributions of the 1983, 1984 and 1985 mortality estimates. For 1985, the distribution of the estimate consisted of the 1,000 values from the mortality bootstrap. Because the 1983 and 1984 estimates were not computed by the stratified bootstrap method[8], their distributions were assumed to be normal, with the standard deviations given above. The central 95 percent of the bootstrapped regression slopes are shown by dashed lines in Figure 5. The fitted mortality estimates for the fishing seasons 1968-1985, are given in Table 5. Those estimates are used in the analysis presented in the section entitled OSP DETERMINATION.

If the harbor seals in the Southern California Bight were considered a separate stock from the remainder of California, the stock size would be estimated from the count at the Channel Islands plus the mainland count for sites south of Point Conception. The 1985 count for those areas was 3,650 seals (Hanan et al. 1986b, Stewart et al. in press). Applying a correction factor of 1.4 yields a stock size estimate of 5,100. The 1985

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[8]Personal communication. D. Hanan, California Department of Fish and Game.

kill estimate for fisheries in the SCB was 739 harbor seals (Hanan et al., in prep.), or about 15 percent of the estimated stock size.

OSP DETERMINATION

California

As discussed in the section on methods of population assessment, a strict application of dynamic response analysis to harbor seals would require including information about the annual levels of incidental mortality. In the method presented by Goodman (in press) per-capita production is estimated by some function of the first difference of the time series of abundance estimates and the harvest. The particular function used depends on the relative timing of the incidental mortality, the census, reproduction, and natural mortality. The estimated per-capita production is then smoothed by fitting a relationship between per-capita production and the associated stock sizes. Finally, a "local" analysis of the slope of the relation between per-capita production and stock size is used to determine the qualitative status relative to the MNPL.

Figure 6 shows per-capita production estimates plotted against stock size estimates from mainland California censuses corrected by the central California mortality estimates in Table 5. There is no indication of a monotonic decreasing function in Figure 6, probably due to the sensitivity to noise in the data when forming ratios of production to abundance. Uncertainty in the counts, in correcting for hauling patterns, and in the mortality estimates ensures that the coupling between abundance, production, and incidental mortality is very weak.

Previous applications of dynamic response analysis (DeMaster et al. 1982, Boveng et al. in press) have avoided the necessity to directly estimate per-capita production by considering only unharvested populations and time intervals without obvious large scale environmental disturbances. Under those conditions, the analysis may be performed simply by considering the shape of the trajectory of abundance estimates, thereby reducing the sensitivity to some types of noise in the data (Goodman, in press). The set of harbor seal counts from mainland California (Fig. 2) spans an interval during which there was a substantial incidental kill as well as a major environmental disturbance. However, applying this simplified dynamic response technique to the counts obtained prior to El Niño could be informative, despite the existence of the incidental take, if the analysis indicated the stock was clearly below its MNPL. In fact, if a stock subject to a non-decreasing take was determined to be below MNPL by the method which ignores the take, then the actual status would be even farther below MNPL than the apparent status. If, on the other hand, the analysis revealed a stock at or above MNPL, it

would be impossible to distinguish, from the population index alone, between density dependent changes in the growth rate and the effects of the incidental mortality.

A second dynamic response analysis of the mainland California harbor seal counts was conducted by least squares fitting of a second degree polynomial function of time (independent variable) to the counts from the years 1965 through 1982. The series of 7 counts from that time interval (Fig. 2) is too short to use the moving interval technique (Boveng et al. in press) to estimate the minimum number of censuses needed to reliably detect density dependent curvature. However, the sign of the second order coefficient from that single interval is positive (coefficient = 4.91, s.e. = 2.54, P<0.10). At a level of significance of .10, this result would be taken to indicate population dynamics (average, during the interval) like those expected from a population below MNPL. At a level of .05, the result would be ambiguous, for the reasons given in the previous paragraph. Therefore, although the trajectory of harbor seal counts from mainland California (Fig. 2) prior to 1983 appears to represent an increasing growth rate characteristic of a population below its MNPL, the data are too uncertain to make a definitive statement.

Because the data from the Channel Islands are subject to many of the same uncertainties (short time series, El Niño, uncertainty in correction factors, few mortality estimates) as the mainland data, the status of the island seals is also uncertain.

Other Stocks

Since the passage of the MMPA in 1972, harbor seals have begun to occupy several new sites in Oregon and are not known to have abandoned any sites there (Brown 1986). Brown (1986) suggested that the expansion may be due to decreased harassment of seals on breeding and pupping sites, particularily estuaries and bays. It is not known whether the apparent expansion represents population growth or redistribution or both. Thus, the implications for population status are not clear.

Because the quantitative data on stock sizes and trends, and on incidental mortality are probably more uncertain for all other stocks than for California, rigorous determination of status relative to the optimum sustainable population levels is not presently possible.

REPLACEMENT YIELD AND MAXIMUM ALLOWABLE TAKE

Replacement yield is defined here to be the level of take which would result in a stationary population at the current population size. An estimate of replacement yield is needed to establish levels of maximum allowable take. For a stock found to

be within its range of OSP, the replacement yield could be estimated if sufficient information about observed growth rates or potential rates derived from life history parameters was available. (A realistic estimate would need to be expressed in terms of the age and sex structure of the take, as well.)

For each of the stocks considered in this paper, there is sufficient uncertainty about the current stock sizes, growth rates, and population structures, that meaningful estimates of replacement yield cannot be derived. However, in two prior cases when estimates of replacement yield were not available (bottlenose dolphins - Powers[9], harbor porpoise - Barlow 1987), the "2-percent rule" was recommended as a means for establishing maximum allowable takes. The 2-percent rule simply says that, considering vital rates of most marine mammals, an annual take of less than 2 percent of the stock size should allow a stock to grow and recover even if, because of uncertainty in the data, the stock is incorrectly judged to be above the minimum value for OSP. Applying the 2-percent rule to harbor seals may require more secure estimates of correction factors for the counts, as well as regular adjustments of the annual numerical quotas to account for changes in stock size (Barlow 1987).

Using the intermediate stock size estimates (Table 2, correction factor of 1.4), and the 2-percent rule, the maximum allowable take for the inland waters of Washington would be about 170 animals if the take were distributed in proportion to the population's sex and age structure. That value is at the upper end of the range of estimated incidental kill in recent years for that stock. In the same fashion, for Oregon and the outer coast of Washington, the estimated maximum allowable take would be 330 animals, about 170 animals fewer than estimated current level of mortality. For California, the estimated maximum allowable take would be 498 seals, substantially smaller than the estimated incidental mortality of 1,849 in 1985.

SUMMARY: STATUS OF THE STOCKS

Two assumptions should be noted before proceeding to a summary of the analyses for each stock. First, there was insufficient information to define stocks on a strictly biological basis, so the stock divisions used here are based in part on convenience and management considerations. Second, some of the results are dependent on the factors used to scale harbor seal

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[9]Powers, J. E. (ed.) 1984. Report of the working group on marine mammals. pp. 68-90 In Report of the Second Southeast Fisheries Center Stock Assessment Workshop. Manuscript on file, National Marine Fisheries Service, Southwest Fisheries Center, La Jolla, CA.

counts to estimates of total population sizes. I have assumed that total population size is between 1 and 2 times the count at peak seasonal haulout, with a factor of 1.4 considered most likely. Other interpretations of the studies of hauling patterns (reviewed above) are possible.

Inland waters of Washington

The recent increases in several indices of harbor seal abundance observed by Calambokidis et al. (1985) and Jeffries[10] suggest that the levels of take in that stock have been below the maximum allowable take. There is little information which bears on the question of stock status relative to OSP. The current level of take (estimated at 50 - 100 animals per year) appears to have a negligible impact on the stock.

Washington and Oregon

Combined counts and estimates from censuses of major portions of this stock indicate that it comprises at least 11,700 seals. However, some of the counts and estimates are several years out of date. As more recent counts become available, stock size and status estimates for this stock will be more secure. Trends in statewide counts from Oregon and from Willapa Bay and Grays Harbor indicate that the stock was increasing, at least until 1982. The time series of counts are not sufficiently long to rigorously determine status relative to OSP. The estimated level of incidental take by commercial fisheries lies between 2.1 and 4.2 percent of the estimated stock sizes. The recent level of incidental mortality does not appear to have significantly impacted the stock, but again, both the mortality and the stock size estimates are based on data which are several years old.

California

In June of 1986, there were at least 17,800 harbor seals in California waters. On the basis of several published and unpublished studies of hauling patterns, I have assumed that the actual population size is about 1.4 times the number counted, or about 24,920. The apparent rates of growth of this stock prior to 1983 were high, possibly 15 percent annually (but the apparent growth rates may be inflated by progressively improved census technique). Counts declined in 1983, almost certainly in response to El Niño. Subsequently, the counts have increased each year, but the time series is too short, and the relationship between

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[10]Personal communication. S. Jeffries, Washington Department of Wildlife.

counts and total stock size too uncertain to determine whether the increasing counts represent genuine population growth.

The status of the stock relative to its MNPL has not been satisfactorily determined. A crude dynamic response analysis, utilizing a method for unharvested populations, revealed little more than could be inferred from observation of the shape of the curve in Figure 2. The appearance of that curve suggests that the stock was, on the average, below its MNPL between 1965 and 1982. The recent counts have not increased significantly above the 1982 level, making it unlikely that the status has changed substantially.

There is no firm evidence to suggest that the stock is above its MNPL. The counts since 1982 show no significant trend, but the occurrence of El Niño and the possibility that incidental fishery mortality has increased, are plausible explanations for the disruption in the growth curve. The increases in counts in 1984, 1985 and 1986 may represent continued population growth, but the time series is too short to adequately assess statistical significance.

The population dynamics of this stock in the near future will likely be affected by two significant recent events. First, the sea surface temperature anomaly (El Niño) experienced in California waters beginning in 1983 is known to have affected populations of many species of animals (Barber and Chavez 1984). It is likely that survival and reproduction of harbor seals were altered during El Niño (Stewart et al., in press) and that some of the effects persisted for several years owing to the age structure of the population. Second, the level of gill net fishery effort increased substantially in the late 1970's. There is little information bearing on the ability to distinguish between the effects of El Niño and the effects of incidental mortality on the stock.

ACKNOWLEDGEMENTS

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Table 1. Minimum population estimate for harbor seals on the west coast of the continental U. S.

Region	Count	Date
California	13,913	1986
Channel Islands	3,887	1986
Oregon	3,825	1984
Willapa Bay and Grays Harbor, WA	5,869	1982
North Coast, WA	2,103	1978
Inland Waters, WA	6,062	1984
Total	35,659	

Table 2.   Minimum harbor seal stock size estimates, with total stock size estimates based on correction factors of 1.4 and 2.0.

Stock	Minimum Estimate	Correction Factor = 1.4	Correction Factor = 2.0
1) Inland Waters of Washington	6,062	8,487	12,124
2) Washington and Oregon (outer coast)	11,797	16,516	23,594
3) California	17,800	24,920	35,600
TOTAL	35,659	49,923	71,318

Table 3. Estimates of fishing effort and harbor seal mortality incidental to California fisheries in 1985, stratified by area and several types of fishery (from Hanan et al., in prep.).

Stratum	Estimated Total Effort (net pulls)	Estimated Kill (number of seals)
SAN FRANCISCO AREA		
Halibut/Shark/Flounder:		
Half Moon Bay		
soak time < 48 hours	469	56
soak time ≥ 48 hours	457	190
Bodega and S. F. Bays		
soak time < 48 hours	1,858	92
soak time ≥ 48 hours	226	23
MONTEREY		
Halibut/Shark/Flounder:	1,255	178
MORRO BAY		
Halibut/Shark/Flounder:		
July-September		
depth ≤15 fathoms	808	340
depth >15 fathoms	579	63
Remaining months		
depth ≤15 fathoms	1,122	91
depth >15 fathoms	954	77
SOUTHERN CALIFORNIA		
Halibut and angel shark:		
Channel Islands	5,319	380
mainland	18,425	115
Soupfin shark fishery:	2,275	92
DRIFT GILL NET FISHERY	10,000	152
TOTAL		1,849

Table 4.   Estimated 1985 harbor seal mortality incidental to
California fisheries, presented as proportion of 1985 stock size
under three different assumptions about correction factors for
counts.

Correction Factor	Stock size estimate	Kill As Proportion of Stock Size < 90 percent confidence interval >		
		lower bound	mean	upper bound
1.0	15,585	.090	.118	.144
1.4	21,819	.065	.084	.103
2.0	31,170	.045	.059	.072
Kill in numbers of animals:		1,410	1,849	2,250

Table 5.   Central California incidental harbor seal mortality estimated by regression (through the origin) of the CDFG direct estimates on number of halibut landing receipts attributed to set nets.

| Fishing Year | Landing Receipts | Regression Estimate | | | CDFG Direct Estimate |
		95% Lower Bound	Mean	95% Upper Bound	
1968/69	70	24	30	37	
1969/70	256	88	111	134	
1970/71	190	65	82	100	
1971/72	419	144	182	220	
1972/73	565	193	245	297	
1973/74	178	61	77	94	
1974/75	353	121	153	185	
1975/76	696	238	302	366	
1976/77	869	298	377	457	
1977/78	676	231	293	355	
1978/79	1131	387	490	594	
1979/80	1629	558	706	856	
1980/81	2053	703	890	1078	
1981/82	2573	881	1116	1352	
1982/83	2015	690	874	1059	
1983/84	1894	649	821	995	567
1984/85	2266	776	983	1190	857
1985/86	1788	612	775	939	1110
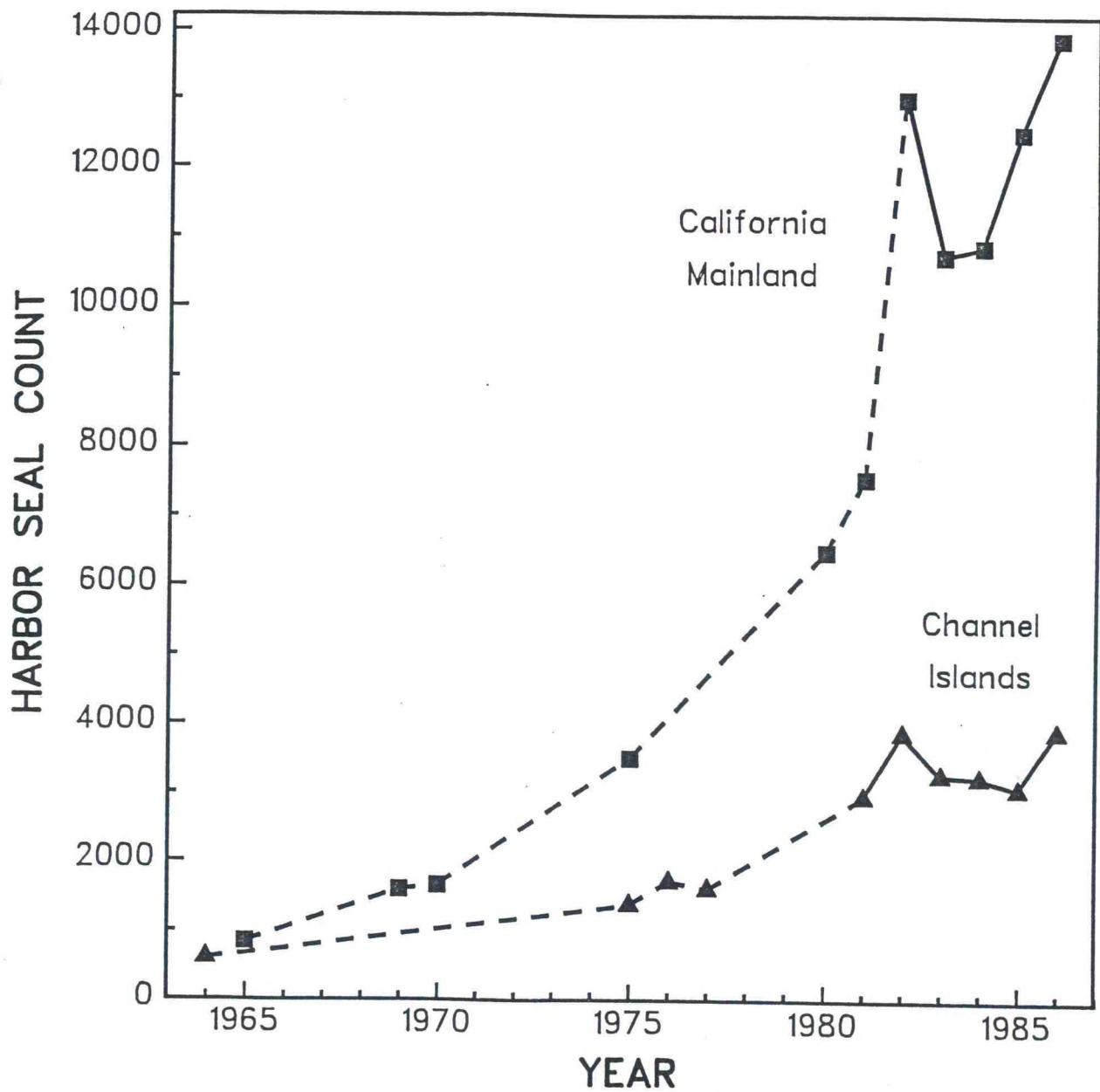
Figure 1. Harbor seal counts from mainland California and the Channel Islands. Dashed lines indicate that the counts were obtained by several techniques which may not be comparable. Solid lines connect counts which were collected by consistent technique. Data sources are cited in the text.
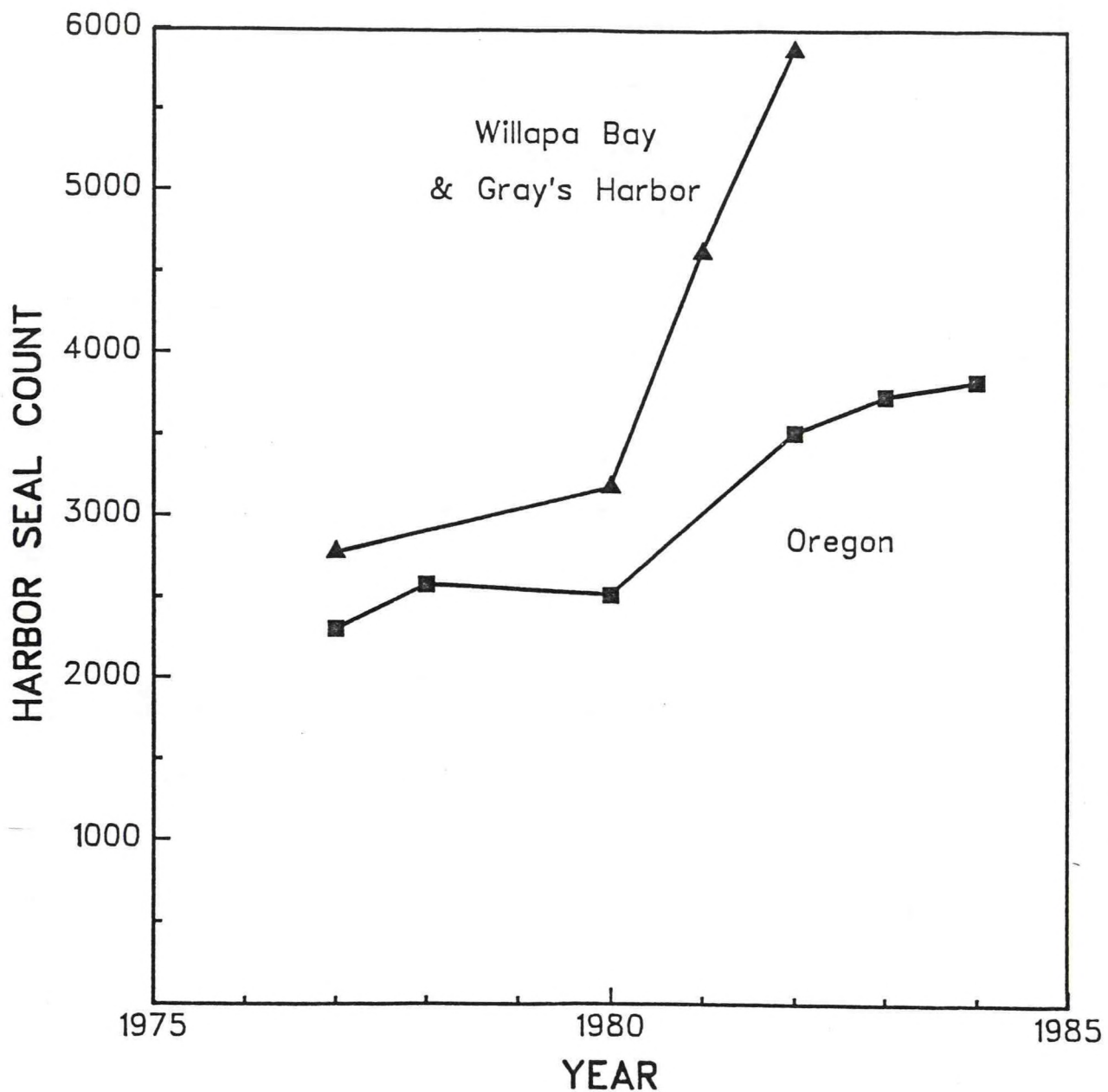
Figure 2. Harbor seal counts from Oregon, and from Willapa Bay and Gray's Harbor, Washington. Data sources are cited in the text.
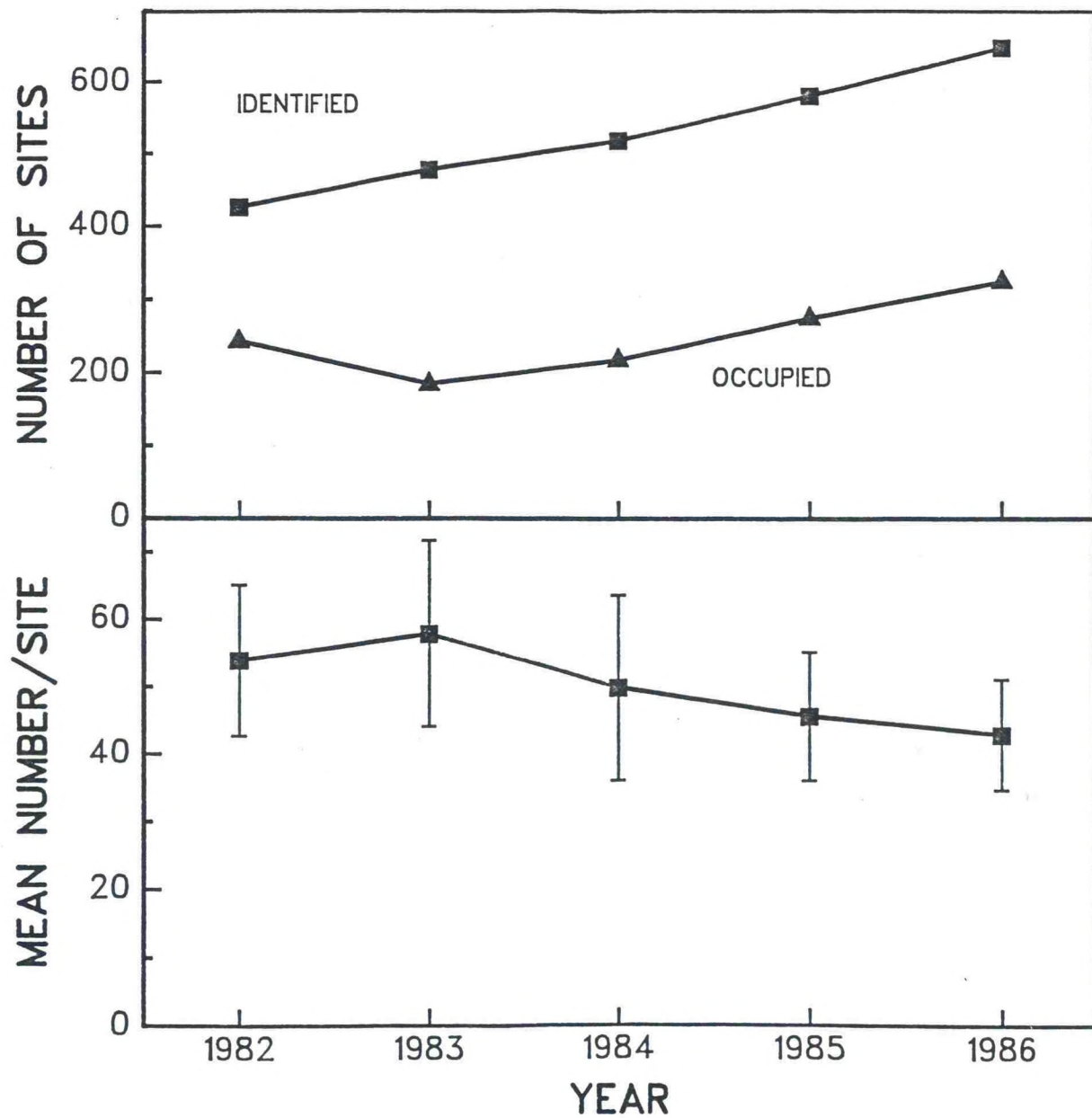
40

Figure 3. Summary statistics from CDFG harbor seal surveys of mainland California. The upper figure shows the cumulative number of hauling sites identified, and the number of those sites found occupied by seals each year. The lower figure shows the mean number of seals per site. Error bars are ± 2 standard errors of the mean.
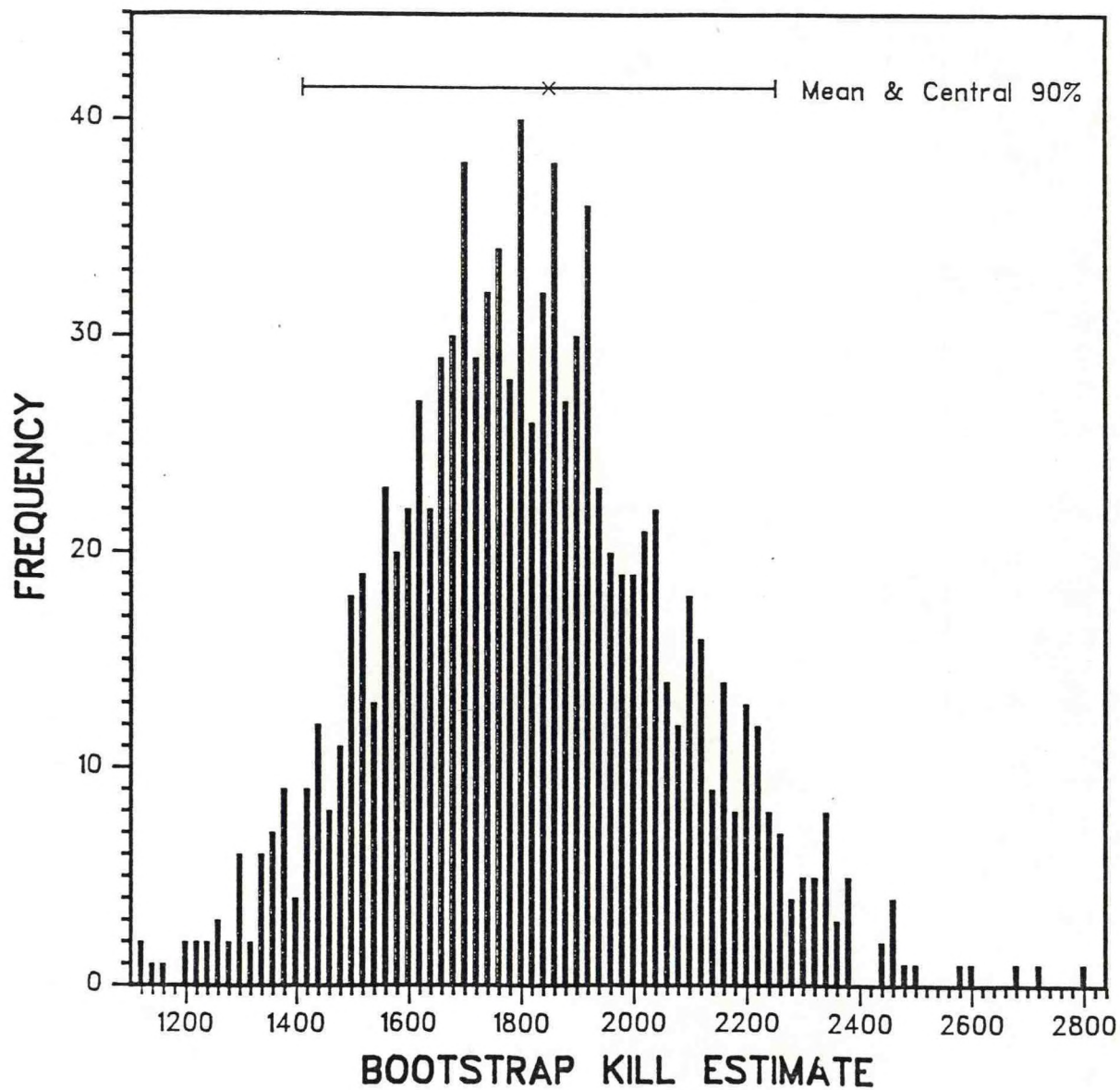
Figure 4. Distribution of incidental mortality estimates obtained by bootstrap resampling of distribution of observed kills in California gill nets.
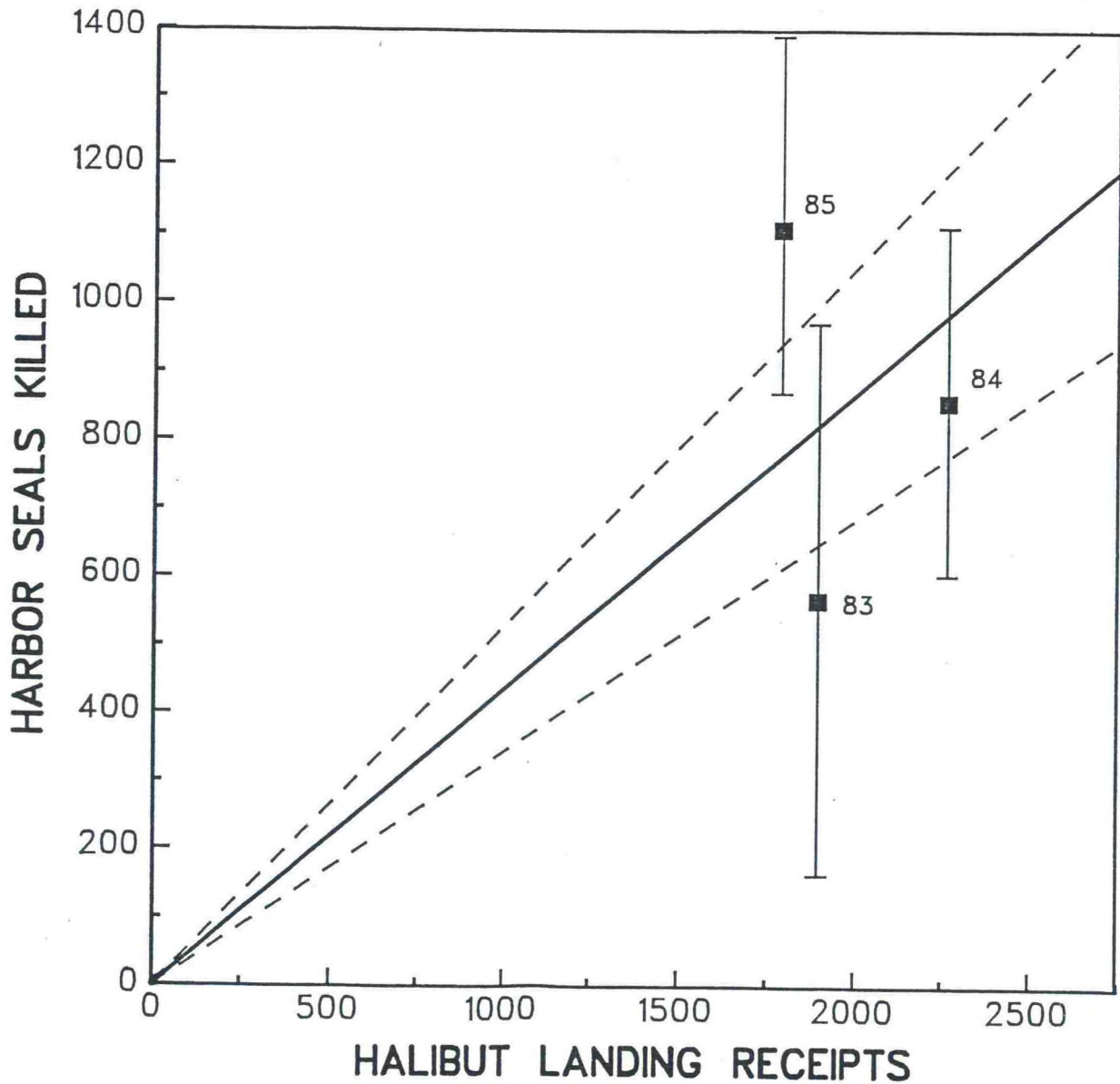
Figure 5. Relationship between halibut landing receipts and estimated harbor seal mortality incidental to gill net fisheries in central California, 1983, 1984 and 1985. The regression assumes that zero halibut landings would result in zero mortality. Dashed lines are approximate 95% confidence limits.
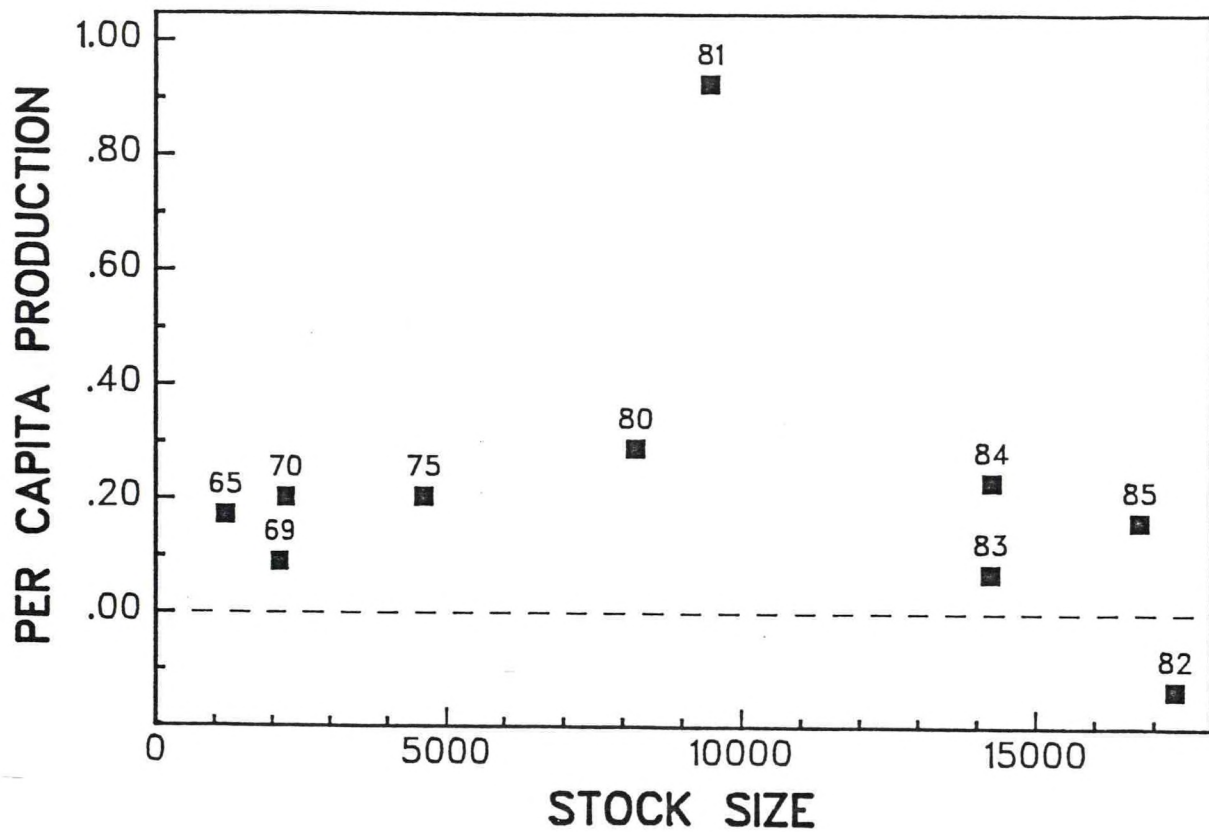
Figure 6. Per-capita production estimates plotted against stock size estimates from mainland California censuses corrected by the central California mortality estimates in Table 5.